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Biological Diversity: Status and Trends in the United States

Linda L. Langner and Curtis H. Flather

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Biological Diversity: Status and Trends in the United States

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INTRODUCTION

The Nature Conservancy (1975) issued a report on natural diversity, almost 20 years ago, that indicated a rapid loss of ecosystems and communities in the United States. The report emphasized the reliance of society on ecological systems, and the need to protect natural areas that support species and natural communities. Today the same concerns are being expressed in terms of biological diversity.

This report summarizes information available that can provide evidence about the status and trends in biological diversity in the United States. A literature review of the topic has been handled elsewhere (Myers 1983, Wilson 1988a, Wilson 1992).

DEFINITION OF BIOLOGICAL DIVERSITY

Diversity is a complex quality that includes both the number and relative frequency of biological entities. Measures of diversity should include both of these features. Peet (1974) argued that, because there was no consensus on the best way to measure diversity, it was difficult to clearly state ideas and hypotheses about diversity. Patil and Taillie (1982) suggested that a consensus will be difficult, despite extensive literature. Decisive definitions of diversity that would guide the development of operational approaches to conservation of diversity still are lacking (Samson and Knopf 1994). Despite the problems of semantics, there is no disagreement that reducing the number of biological entities in a system or making some of them less abundant reduces diversity.

This report adopts the definition from the Keystone report (1991:6): "the variety of life, and its processes; including the variety of living organisms, the genetic differences among them, and the communities and ecosystems in which they occur." This definition incorporates the concept of interactive "levels" of diversity. The three levels commonly identified are genetic diversity, species diversity, and community or ecosystem diversity.

Genetic Diversity

Genetic diversity is the variability of genes, calculated from the number and frequencies of alleles and their combinations, among individuals in a species or population. A species' evolutionary ability depends on sufficient genetic diversity to maintain immediate fitness and adaptability (Soulé 1980), which, in turn, are a function of population size, subpopulation structure, interpopulation genetic variation, mating structure, generation time, and gene flow (Frankel 1974; Franklin 1980; Namkoong 1983).

Concern about genetic diversity is most serious for populations that are either naturally small and isolated, or populations that have become small because of changes in their environment. Consequently, population size (Lande and Barrowclough 1987) and the distribution of interacting populations (Gilpin 1987) are critical attributes used to evaluate genetic diversity. If populations decline to critical levels and become isolated, inbreeding (mating among relatives) can reduce genetic variation and increase the frequency of lethal alleles, even over a period of several generations. The probability of losing rare alleles increases during such episodes. These rare alleles may be important for the species to survive in special environmental circumstances. Loss of genetic diversity also is associated with increased mortality rates, reduced fecundity, increased incidence of birth defects, and reduced resistance to disease, a pattern referred to collectively as inbreeding depression (Lacy 1992). Inbreeding ultimately threatens the persistence of the original population. The process whereby allele frequencies change by chance (i.e., the random sampling of gametes during mating) is called "genetic drift." In small, isolated populations, genetic drift gradually reduces genetic diversity.

The relationship between genetic diversity and population persistence can be analyzed using population viability analysis. Population viability analysis examines the influence of random processes on the persistence of small populations, rather than

systematic pressures (e.g., habitat loss, exploitation) that normally receive resource management emphasis (Brussard 1991). Shaffer (1981) proposed four random processes that can affect viability: (1) demographic — intrinsic random variation in birth and death rates; (2) environmental — variation in birth and death rates attributed to changes in habitat, interspecific interactions, and disease; (3) genetic — random events that are associated with changes in allele frequency caused by inbreeding or genetic drift; and (4) catastrophic — random incidence of devastating environmental events, such as fire or drought. Within the context of these random processes, population viability analysis predicts the probability that a population of a certain size will persist for a specified period of time. Because populations require space (habitat), it is important to identify those habitat characteristics (quantity and quality) that will support the specified viable population (Gilpin and Soulé 1986).

Species Diversity

Species diversity is a function of both the number of species (referred to as richness) and the proportional number of individuals within each species (referred to as abundance or evenness) in a defined area or group of organisms. Species collections tend to be characterized by many species that are relatively rare, and few species that are very common (Hairston 1959; Hughes 1986). This qualitative generalization notwithstanding, species assemblages from different ecological communities tend to have varying patterns of commonness and rarity (Kilburn 1966).

Whittaker (1960, 1977) developed a hierarchical typology of diversity measures that was keyed primarily to the geographic scale of measurement. Although seven measures were eventually proposed (Whittaker 1977), three are commonly referenced: alpha, beta, and gamma diversity. Alpha diversity is within-habitat diversity, or the number of species in a particular habitat. Beta diversity is the between-habitat diversity, or species that are added when additional habitats are sampled. Gamma diversity is the number of species in a large landscape containing many different habitats (Peet 1974, Cody 1975, Whittaker 1972).

Within this hierarchy, several diversity indices have been developed, each representing an attempt to quantitatively distinguish among species collections. Although the profusion of metrics has contributed to the criticism of diversity indices, there are valid concerns about the interpretability of a single number that represents the occurrence and abundance among all species in a collection (Hurlbert 1971). Comprehensive discussions of diversity indices are in Pielou (1975) and Magurran (1988).

Community and Ecosystem Diversity

Community and ecosystem diversity reflects the kinds and amount of community/ecosystem types across a geographic area. Communities generally are described as a group of interacting plants and animals in an area, and frequently are defined by their floristic and faunal similarity. Often, communities are delineated as plant associations, such as deciduous forests, tall-grass prairies, or sedge meadows. Ecosystems encompass these communities, but include both biotic and abiotic factors that are united by the exchange of energy or matter (MacMahon et al. 1981). While community definitions focus on species composition, ecosystem classifications include structural and functional attributes, such as the ability to capture and transfer energy and nutrients (Solbrig 1991).

The two terms often are used somewhat interchangeably in discussions of biological diversity, and often in the context of mappable entities distributed across a landscape. This latter, restricted aspect of community and ecosystem is measured to evaluate community/ecosystem diversity. Community/ecosystem diversity can be characterized according to the kinds and relative areas of community or ecosystem types within a region. Consequently, simple measures of land type composition, and more complex attributes reflective of the configuration and spatial arrangement of land types, all qualify as measures of community or ecosystem diversity. Examples of spatial attributes include degree of fragmentation, connectedness, patch shape, edge complexity, and average patch size, attributes that are associated with measures of landscape structure (O'Neill et al. 1988; Turner 1990b; LaGro 1991).

THE GLOBAL CONTEXT FOR BIOLOGICAL DIVERSITY IN THE UNITED STATES

One of the most commonly recognized macroecological patterns of biological diversity is the increased richness of the biota as one moves from the poles to the tropics. Wilson (1988b:8) estimated that, although tropical habitats only comprise 7% of the global land base, they support more than 50% of the species. This pattern appears to be consistent among various taxa. Tropical countries support more species in almost all taxonomic groups. For example, Brazil and Colombia both have over 50% more avian species than the United States, and more than twice as many plant species (table 1). The difference in species richness is even more striking in comparison to other temperate countries (table 1), because the United States has considerably more species diversity than most other temperate countries.

Varying area among countries can confound interpretation of simple measures of species richness. However, Reid and Miller (1989) found the latitudinal gradient in diversity is maintained even after controlling for differences in total area. For example, the United States supports approximately 2,000 vascular plant species/10^4 km^2, compared to 6,000 species/10^4 km^2 in Brazil. Mexico and Zaire support 108 and 96 mammal species/10^4 km^2, respectively, compared to 60 species/10^4 km^2 in the United States. The species richness of the tropics is the primary reason why those areas dominate discussions of biological diversity. However, biotic integrity is independent of the absolute species richness of any given geographic region.

Biotic integrity is not easily quantified (Karr 1992); yet, a simple indicator of integrity can be inferred by looking at the percentage of species considered threatened by extinction. The World Resources Institute (1992) compiled data on the number of threatened species, by taxonomic group, in all countries. The percent of threatened species in a sample of temperate and tropical countries are displayed in table 2 for mammals and birds and plants. Except for the high percentage of threatened mammals and birds in Madagascar, the percent of threatened mammals and birds is not greatly different between temperate and tropical countries. The threat to plant species appears to be greater in the temperate countries. However, it is important to be careful in interpreting these data, because of the differences among countries in efforts made to identify threatened flora and fauna. Until standard criteria for identifying species vulnerable to extinction can be implemented worldwide, this will continue to hamper assessments of species rarity.

Protection of natural areas is a key component of protecting biological diversity. An inventory of protected areas in 1990, compiled by World Resources Institute (1992), indicated that the United States had the most acres set aside in natural areas, about twice as large an area as the country with the second largest protected acreage. Twenty countries had a higher percentage of their land base in protected areas. Ecuador had the highest percentage (38%), compared to 10.5% in the United States. Worldwide, 4.8% of the land base was classified as protected areas.

Table 1.—Species diversity by taxonomic group and country.

Country	Mammals	Birds	Reptiles	Plants
United States	466	1090	368	20,000
Canada	197	426	42	3,220
Spain	108	344	64	4,900
Mexico	439	961	717	20,000
India	341	1178	400	15,000
Colombia	358	1665	383	45,000
Brazil	394	1567	467	55,000

Source: World Resources Institute (1992)

Table 2.—Comparison of degree of threat to mammal and bird and plant species among selected temperate and tropical countries.

Country	Mammal & bird species	Percent threatened	Plant species	Percent threatened
Temperate				
United States[1]	1556	4%	20,000	12%
Canada	623	2%	3,220	<1%
Spain	452	6%	8,937	11%
Japan	856	4%	4,022	17%
Argentina	1182	7%	9,000	<1%
Tropical				
Brazil	1961	7%	55,000	<1%
Colombia	2023	5%	45,000	<1%
Madagascar	355	23%	12,000	2%
Indonesia	1979	9%	32,600	<1%
Zaire	1495	3%	11,000	<1%

Source: World Resources Institute (1992)

[1]*The number of threatened species listed for the U.S. is less than currently on the federal threatened and endangered species list.*

However, size is not as important as which elements of diversity are being protected. Also, the degree of protection afforded to these "protected areas" can vary greatly.

STATUS AND TRENDS IN BIOLOGICAL DIVERSITY

Ideally, data on biological diversity should describe both the distribution and abundance of components of diversity over time. However, information about the status and trends of biological diversity is generally of two types. The first involves direct measures, such as population trends or species richness. The second type involves surrogate indicators that probably are correlated with trends in biological diversity. This report uses a combination of direct and correlated trend information to describe diversity for each of the three levels described previously. Much of the data are of limited scale and generalizability. However, the accumulation of small-scale evidence, from different geographic regions, can provide indications of large-scale patterns.

Other than simple, descriptive statistics reflective of trends in biological diversity, determining the causes of observed trends is more difficult. Two broad categories of causation are important: natural causes (e.g. succession, natural catastrophe) and human-induced causes (e.g. habitat alteration). In many cases, there are multiple, often interactive, factors that affect diversity. Although the relative contributions of natural and human-induced factors are not easily separable, evidence indicates that human-induced factors are the primary cause of concern.

Status and Trends in Genetic Diversity

Limited information is available to assess genetic diversity systematically over broad geographic regions within the United States. Available studies were grouped into two categories. The first is genetic structure, which includes studies about genetic variation within and across individuals. The second category is population viability, which includes studies focusing on how population size and distribution affects genetic variation.

Genetic Structure

Studies of the genetic structure of individuals or populations of species are relatively scarce, because techniques for detecting variation are new and difficult. Our understanding of speciation, adaptation, and evolution will increase as more information on the genetic structure of species becomes available. Loveless and Hamrick (1984) found such information scarce in plants. There appears to be even less information on animal species, particularly wild species.

Initial concern about genetic diversity focused primarily on the genetic erosion of major crop species in the United States. Questions were raised about the ability of existing gene pools to meet future human needs for food and fiber (National Research Council 1978, Namkoong 1981). Detailed examination of the status of genetic resources for crop breeding is in Oldfield (1984). Genetic resources for continued improvements in agricultural crops continue to be important. The wild relatives and progenitors of almost all major U.S. crops are found outside the United States; but, the continued existence of these genetically diverse wild relatives is very important. Native food crops, such as cranberry (*Vaccinium macrocarpon* Ait. and *V. oxycoccus* L.), and sunflower (*Helianthus annus*), are of minor commercial value relative to introduced species, such as corn *(Zea mays)* and wheat *(Triticum aestivum)* (Oldfield 1984).

Wild species, particularly plants, are also the source of many medicinal compounds. Systematic efforts to screen species for medicinal properties are now underway in the U.S. and abroad. An example in the U.S. is the Pacific yew *(Taxus brevifolia),* a species considered a weed tree until its use as a source of taxol was discovered. A draft environmental impact statement, published for public review in January 1993, discussed alternatives to allow harvest of Pacific yew, balanced with management to protect the viability of the species for future use. (USDA Forest Service, USDI Bureau of Land Management, and USDHHS Food and Drug Administration 1993).

Forest trees have been a second focus of genetic study. Like agricultural crop species, the motivation for most tree genetic studies has been to breed species with improved timber productivity. Unlike crop species, the species of interest are primarily native to the United States. Although many conifer species have been subjected to genetic analysis, the results are not easily generalized to other species. Forest tree

populations vary greatly. Some species have greater genetic variation within populations; other species have greater variation between populations (Namkoong 1981). For example, loblolly pine *(Pinus taeda)* is widespread and highly variable; red pine *(Pinus resinosa)* is widespread and genetically depauperate (Ledig 1988).

The consensus seems to be that forest trees have not been subject to the same degree of loss of genetic variation as agricultural plants and domestic animals, which have been selectively bred for a much longer period of time (Namkoong 1981). However, genetic diversity may have been diminished in areas where large areas of forests were cleared for agriculture. Similarly, logging practices that removed the "best" trees may have selectively removed certain alleles from the gene pool of some species (Ledig 1988).

Genetic studies of wild animal species are rare. Those that exist generally focus on endangered species, which can be expected to have reduced genetic variability. Wayne et al. (1991) studied the genetic variation in the gray wolves *(Canis lupus)* of Isle Royale, a small isolated population in upper Michigan. The results indicated a 50% decline in allozyme heterozygosity, a pattern consistent with predictions for isolated populations with a small effective population size. Stangel et al. (1992) surveyed the genetic variability of the red-cockaded woodpecker *(Picoides borealis)*, whose population has been fragmented by land-use patterns. Tests of heterozygosity in different populations indicated that, although heterozygosity was reduced in some of the small populations, those small populations were seen as important reservoirs of unique genetic combinations and also as a source of gene flow among the larger populations.

Population Viability

The status of genetic diversity can be inferred by the number and size of populations, particularly within protected reserves. Libby et al. (1975) studied conifer gene pools in California, focusing on commercially important species. The study concluded that few entire species or subspecies were considered in danger of extinction in California. However, only two of nine commercial species were considered to have sufficient populations and distribution to maintain genetic diversity. The major gap in representation was populations in mid-elevational areas. High-elevational populations were adequately represented in wilderness areas, and low-elevational populations were included in parks. The study also concluded that continued urbanization and contamination of native gene pools by artificial regeneration practiced within the species' native range threated genetic diversity. Riggs (1990) analyzed the distribution of 85 principal forest tree species in California, on reserves in various ownerships. He concluded that the distribution of species across the existing reserve system may not protect the genetic diversity of 30% of the forest trees in California.

A concern about contamination of native gene pools also was raised with salmonids in the Pacific Coast Region. The four-state region of California, Washington, Oregon, and Idaho have 214 stocks of Pacific salmonids that have been identified as of special concern. Hybridization of wild and hatchery stocks is threatening the viability of salmon populations, because of declining genetic vigor observed in hybridized stock (Meffe 1992).

Berger (1990) looked at small, insular bighorn sheep *(Ovis canadensis)* populations to see how long they resisted extinction. Populations of 100 or more sheep persisted up to 70 years; those with fewer than 50 individuals became extinct in less than 50 years. None of the potential causes of rapid population loss (predation, food shortage, climatic severity, interspecific competition) appeared related to these population trends. Therefore, the results were believed to provide evidence of the influence of population size on population persistence time for bighorn sheep.

The northern spotted owl *(Strix occidentalis caurina)* has received much recent viability investigation. Although the conclusions on population persistence times from these independent efforts (Marcot and Holthausen 1987, Lande 1988, Lamberson et al. 1992) vary, primarily because of differences in methodology and assumptions, there appears to be consensus that persistence of spotted owls is a matter of demographics instead of genetics. Adult and juvenile survival and juvenile dispersal appear to be the demographic parameters most sensitive to the removal and fragmentation of old growth habitats. These findings, however, should not be taken as evidence against the importance of genetic variation to population persistence. As Simberloff noted (1987), the information base from which to judge the genetic threat to owl persistence is so inadequate, that there is no justification for disregarding the influence of genetic simplification.

Given the lack of systematic assessments of genetic variation in species within the United States, it is difficult to conclude with any confidence what the general trend in genetic diversity has been in recent times. Local investigations tend to support the predictions of theory, namely, that small isolated populations suffer from a degradation of genetic variation. As the human population within the U.S. increases, and natural vegetation is converted to intensive land uses, more and more species will face situations in which genetic problems are likely. Such patterns indicate that the genome of a species is likely to become simpler in the future for much of the biota, and, therefore, less adaptable to environmental change.

Status and Trends in Species Diversity

Species diversity is an important attribute of ecological systems for which there is a wealth of information. Species diversity clearly diminishes as species become extinct.

Only native species are discussed here. Although many exotic species have been introduced into the U.S., no attempt has been made to account for these additions to native diversity. Exotics often have detrimental impact on native species, especially native species already suffering from other sources of stress. Some introduced species have been a primary or contributing cause to the endangerment and extinction of native species (Pimm 1986). Assessing the role of exotics as additions to diversity or as a factor in declines in native diversity is beyond the scope of this report.

Trends in Extinction

Extinction is a natural process which, when examined over geologic time, has periodically caused major reductions in diversity (Raup 1988, Gould 1989). Extinctions in geologic time, primarily based on fossil evidence at the level of families or genera, have been used to estimate a "background" extinction rate. Although there is no single estimate of background extinction rates, the range of estimates for modern times likely exceeds those over geologic time (Lovejoy 1986). Wilson (1988b:13) estimates that current global rates of extinction exceed background rates by three to four orders of magnitude.

Scientists can account for the extinction worldwide of about 75 mammals and birds between 1600-1900, averaging about one species every 4 years. Between 1900 and 1980, another 75 mammals and birds became extinct, an increase to about one species a year. Current estimates of avian and mammalian extinction rates are one to three species a year. However, rates for all species range from one to three species a day, to the most pessimistic estimates of up to one species an hour (Council Environmental Quality 1980). Even at the most conservative end of the range, scientists agree that current extinction rates exceed speciation rates, so that a net reduction in diversity results (Office of Technology Assessment 1987).

Of the known species that have become extinct worldwide in the past three centuries, at least 50 were higher vertebrate animals native to North America (Council on Environmental Quality 1974). During the past century, 40 taxa, including 27 species and 3 genera, of North American fish have become extinct (not including marine fish or distinct stocks of anadromous fish). Nineteen of those taxa became extinct since 1964 (Cairns and Lackey 1992). At least eight species of mussels and all five species of an endemic subfamily of limpets have become extinct in the southeastern U.S. Ten percent of North American freshwater mussel species have become extinct since 1900 (Hughes and Noss 1992). About a dozen documented extinctions exist for U.S. plant species; but, perhaps 200 additional undocumented extinctions occurred between 1790 and 1975 (Falk 1990).

Island species are especially susceptible to extinction. Hawaii has experienced more extinctions than the continental United States. Since 1850, 85% of the endemic species of Hawaii either have become extinct or have been severely reduced (Vermeij 1986). Half of Hawaii's endemic birds and hundreds of plants and invertebrates have been lost (Keystone Report 1991).

Recent extinctions, however, are not restricted to island biota. Within the past 10 years, seven species and subspecies have been formally removed from the threatened and endangered species list because they are now believed to be extinct: the dusky seaside sparrow *(Ammodramus maritimus mirabilis)*, Amistad gambusia *(Gambusia amistadensis)*, Sampson's pearly mussel *(Epioblasma samsoni)*, Santa Barbara song sparrow *(Melospiza melodia graminea)*, blue pike *(Stizostedion vitreum glaucum)*, longjaw cisco *(Coregonus alpenae)*, and the Tecopa pupfish

(Cyprinodon nevadensis calidae) (U.S. Department of the Interior, Fish and Wildlife Service 1992). This is an underestimate of recent extinctions in the U.S., because some species become extinct before receiving any formal protection. Estimates by 1990 indicated that more than 200 candidate species and subspecies, the majority of which are invertebrate and plant species, are no longer being considered for formal listing because they are believed to be extinct (U.S. Department of the Interior, Fish and Wildlife Service 1989, 1990).

A variety of factors are responsible for modern extinctions. The evidence from known extinctions indicates that biotic factors such as competition, predation, parasitism, and disease rarely cause extinction, especially in continental species, unless a species is already under stress from other factors (Frankel and Soulé 1981). Habitat destruction, overexploitation, and impacts of exotic species are the most important causes of modern extinctions. Species that are particularly vulnerable to extinction are those with low population densities and large individual ranges, large body size (generally correlated with lower reproductive rates and longer recovery times), species with specialized habitat needs, and species that are mutualistic or co-evolved (Vermeij 1986).

Table 3.—Taxonomic comparison of listed and proposed threatened and endangered species.

Taxon	Federally listed[1]	Category 1[2]	Category 2[3]
Mammals	65	7	202
Birds	85	5	54
Reptiles	34	1	54
Amphibians	11	4	50
Fish	91	15	118
Snails	13	27	143
Clams	42	2	59
Crustaceans	10	2	91
Insects	23	9	584
Arachnids	3	1	27
Plants	351	526	1572
Other[4]	0	0	14
Total	728	597	2954

Source: Flather et al. (1994)

[1]*As of August 31, 1992 (USDI Fish and Wildlife Service, 1992).*

[2]*As of 1989-1990 (USDI Fish and Wildlife Service 1989, 1990a). Category 1 includes those species for which sufficient biological evidence exists to support official listing; but proposed rules have not been issued, yet.*

[3]*As of 1989-1990 (USDI Fish and Wildlife Service 1989, 1990a). Category 2 includes species that have some evidence to indicate that listing may be appropriate; but conclusive biological evidence to support issuing proposed rules is lacking.*

[4]*Includes sponges, hydroids, flatworms, earthworms, and millipedes.*

Threatened and Endangered Species

Several sources of information are available on the status of threatened and endangered species. The most comprehensive national sources are the data base of The Nature Conservancy, and the Fish and Wildlife Service's (FWS) federal list of threatened, endangered and candidate species. The FWS's data were used for this discussion. Although legal mandates require a continual evaluation and updating of species that should receive formal protection under the Endangered Species Act (P.L. 93-205), historic trends in species listings do not solely reflect trends in endangerment. The rate of listing is most sensitive to changes in the law, budget constraints, bureaucratic processes, and policy changes that effect how many listings can occur in any time period. As a consequence, the official list has underestimated the number of species that are at risk of extinction. The formation of a candidate list, which includes species being considered for federal listing, partially offsets that underestimate.

Species have been added to the federal list at different rates over the past 20 years. The most rapid increase in species listing occurred in the past 8 years, when an average of 4.2 species a month were added to the list (Flather et al. 1994). The relative listing emphasis given to various taxa also has varied over time. Vertebrate species dominated the list during the mid-1970s, whereas plants and invertebrates now comprise a much greater proportion of the listed biota (table 3).

While the rate of species listing has tended to increase with time, the number of candidate species has remained relatively constant. Of the more than 3,500 candidate species, 59% are plants, 27% are invertebrates, and 14% are vertebrates (table 3). A recurring source of controversy in the species listing is whether threatened subspecies and populations of otherwise healthy species should be considered for federal protection. Wilcove et al. (1993) reviewed all U.S. taxa proposed for listing or added to the list between 1985 and 1991. Of the 492 proposed listings,

19% were subspecies and 2% were populations. The proposed subspecies and populations were primarily from either mammal or bird species.

Apart from trends associated with the formal listing process established by the Endangered Species Act, there are independent evaluations of species endangerment whose criteria for assessing extinction risk are not tied to the Act. Approximately 10% of 20,000 species, subspecies and varieties of higher plants native to the continental U.S. may be in trouble. In Hawaii, nearly 50% of the total diversity of native vegetation has been described as threatened or endangered (Council on Environmental Quality 1974). An independent evaluation of plants in 1988 indicated that about 680 species were critically endangered. Nearly 75% of these species occur in only five states and territory: Hawaii, California, Texas, Florida, and Puerto Rico (unpublished Center for Plant Conservation Survey; December 9, 1988).

Of the 1,033 known freshwater fish species in North America, the International Union for the Conservation of Nature and Natural Resources (IUCN) considered 74 species endangered, 85 vulnerable, 101 rare, and 27 believed extinct (Williams and Miller 1990). The American Fisheries Society listed 364 North American fish taxa (species and subspecies) as either threatened (114 species), endangered (103 species), or of special concern (147 species) (Cairns and Lackey 1992). The Great Lakes had substantial stocks of sturgeons, salmonids, coregonids, and percids that were reduced to remnants by the early 1900s. Four formerly commercial species are extinct, two are endangered, one is of special concern, and two are rare. Thirty-four percent of the native fishes of the upper Colorado River basin are extinct, endangered or threatened. Since 1950, 67% of the fish species from the Illinois River and 44% from the Maumee River have declined or been extirpated (Hughes and Noss 1992). A study on the status of native fish fauna of California used expert knowledge to quantify trends. Of 115 native fish, 7% of the species were considered extinct, 12% were officially listed for protection, 6% needed immediate listing, 17% may need listing soon, 22% showed declining populations but were not yet in trouble, and 31% seemed secure (Moyle and Williams 1990). Recently, 214 native, naturally spawning stocks of Pacific salmon (*Oncorhynchus* spp), steelhead *(Salmo gairdneri)*, and sea-run cutthroat trout *(Salmo clarki)* were listed at risk in California, Oregon, Washington, and Idaho (Cairns and Lackey 1992). These 214 stocks represent 23% of the Pacific salmonids (Hughes and Noss 1992).

The only group of invertebrates with fairly complete information is mussels. Nearly 50% of all U.S. mussels are listed or proposed for listing. Since 1944, the Tennessee River has lost 20% of its mollusk species, and 46% of the remainder are endangered or seriously depleted throughout their range. Forty-eight percent of mussels of the Ohio River drainage are rare, endangered or extinct. Seventy-three percent of the remaining mussel species in North America are rare or imperiled (Hughes and Noss 1992).

Threatened and endangered species of the United States are not evenly distributed across the landscape. Federally listed species are concentrated in 10 regions of the United States, 3 in the Southeast and the remainder in the more arid Southwest. Within these 10 regions, the taxonomic groups are not evenly distributed. Birds are concentrated in the Florida peninsula and central and southern California; mammals are concentrated in coastal Florida, southern California, northern Rocky Mountains, and Appalachians; fish are concentrated in the southwest and southern Appalachians; reptiles are concentrated in Florida and the Gulf Coast. Regions of high endangerment in the eastern United States were associated primarily with forested ecosystems, while regions in the western United States were associated primarily with rangeland (Flather et al. 1994). The endangerment regions in California, the Southwest, and southern U.S. coincide with areas of high projected population growth and human development.

Status of Other Species Groups

Most native U.S. species are neither endangered nor threatened. Data on the status and trends of these species at the regional or national level is restricted primarily to game species, breeding birds, or other groups of special concern. Species population trends date to the 1950s or 1960s, reflecting management of the past three decades. Some of the species hit record low populations in the early 1900s, primarily as a result of commercial harvest. These species included many large mammals, as well as many species of colonial wading birds and shorebirds.

Mammals.—Data on trends in mammal species are available primarily for game species and furbearers. Dramatic shifts have occurred in the distribution

and abundance of many large mammals since colonial settlement, particularly in the eastern U.S. Moose *(Alces alces)*, elk *(Cervus elaphus)*, bison *(Bison bison)*, wolves, mountain lions *(Felis concolor)*, and black bear *(Ursus americanus)* once were widely distributed in the eastern United States. White-tailed deer *(Odocoileus virginianus)* are now the dominant large mammal in the eastern U.S.; elk and bison were extirpated, although elk have been re-introduced in a few areas. Black bear and moose are restricted in distribution; and the gray wolf and eastern mountain lion *(Felis concolor coryi)* are endangered species that occur in small, geographically restricted populations (Matthiessen 1987).

White-tailed deer have increased dramatically in the eastern U.S., since 1900. Population trends since the 1960s show a continued increase in deer populations, and in some parts of the country are considered excessive (fig. 1a). Wild turkeys also have shown large population increases in the eastern U.S., in the past 30 years, a result of restocking programs and favorable landscape changes. Black bear population trends have varied more. Black bears remain relatively abundant; but, their range is restricted to the more inaccessible areas of their former range, because they are less tolerant of human activity in their habitat than are deer or turkey (Flather and Hoekstra 1989).

The western U.S. has a more diverse group of large mammals, including white-tailed deer and mule deer *(Odocoileus hemionus)*, elk, pronghorn *(Antelocapra americana)*, bighorn sheep, mountain goat *(Oreamnos americanus)*, and moose. Deer (mule and white-tailed) populations in the western U.S. declined during the 1970s (fig. 1a). However, populations recovered and have been increasing in the Rocky Mountain region, and have stabilized in the Pacific Coast region. Elk populations have increased gradually since the 1960s (fig. 1b). However, elk once were the most widely distibuted cervid in North America (Boyd 1978), and are now restricted to the western U.S. Pronghorn populations once numbered between 30 and 40 million, but were reduced to 13,000 animals in the 1920s (Yoakum 1978). Pronghorn populations have increased dramatically since that low, and have shown consistent increases over the past 20 years, as a result of regulation of hunting, improved range condition, and increased habitat (fig. 1b)(Yoakum 1978, Wagner 1985).

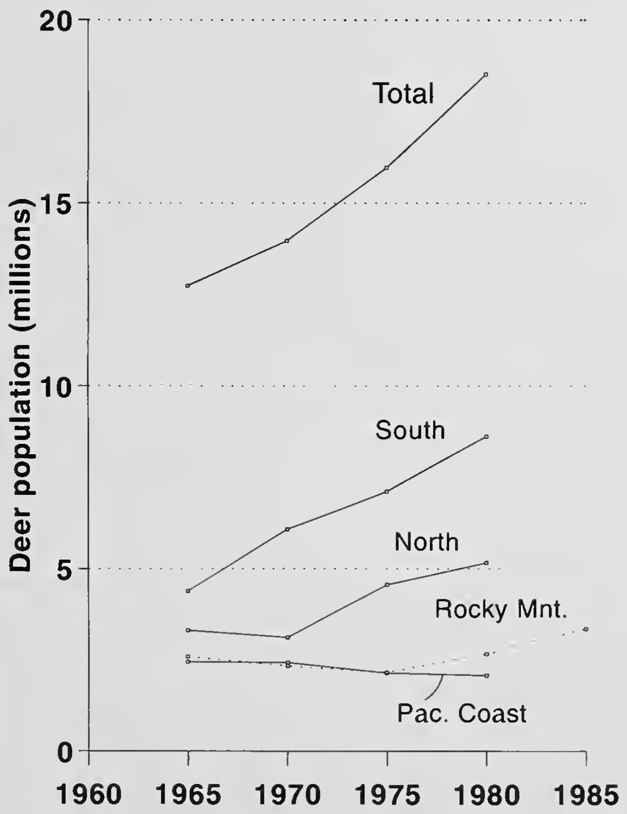

Figure 1a.—Trends in deer populations, 1960-1985.

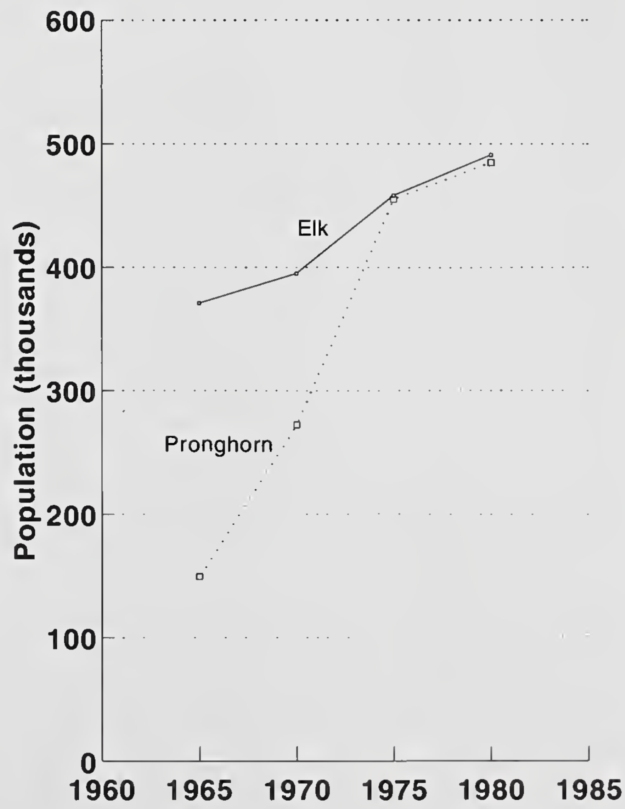

Figure 1b.—Trends in elk and pronghorn populations, 1960-1985

Trend data for small mammals are available for squirrels (*Sciurus* spp.) and rabbits (*Sylvilagus* spp.). Trends depend on whether the species is associated with agricultural land or forestland. The eastern cottontail *(Sylvilagus floridanus)* showed downward population trends between the 1960s and 1980s (fig. 2). Squirrel populations, associated with forestland, declined slightly in the Midwest, and increased in the northeast and southern U.S. (fig. 2a) (Flather and Hoekstra 1989).

Population trends for furbearers, including muskrat *(Ondatra zibethicus)*, Virginia opossum *(Didephus virginiana)*, raccoon *(Procyon lotor)*, beaver *(Castor canadensis)*, mink *(Mustela vison)*, red and gray fox (*Vulpes fulva* and *Urocyon cinereoargenteus*), bobcat *(Felis rufus)*, and coyote *(Canis latrans)*, vary. Muskrat populations continue to be abundant throughout their North American range. Stable populations are expected, with fluctuations generally following wetland habitat conditions (Sisson-Lopez 1979). The raccoon, beaver, and opossum all have shown recent population increases. The raccoon has become more numerous since the turn of the century, adapting well to human populations and extending its range northward. Beaver populations also have increased since the turn of the century, and have expanded back into most of their original range (Deems and Pursley 1983). The opossum remains abundant in the South and also has expanded its range northward.

Fox and mink populations may be declining in some areas of the country. Fox declines may be associated with human pressures in the open prairie regions, while mink declines may be tied to loss of wetland habitats (Sisson-Lopez 1979). Limited information on the bobcat indicated that bobcat populations increased during the 1950s and early 1960s, but have since declined (Anderson 1987). Despite changes in abundance, the distribution of bobcats has changed little historically, except in areas of the midwestern and mid-Atlantic states, where intensive agriculture eliminated populations (Deems and Pursley 1983, Koehler 1987). The coyote appears to be increasing and expanding in many regions despite intensive control programs (Moore and Millar 1984). Coyote range expansion probably is tied to elimination of the gray wolf, clearing of forests, and adaptation to suburban environments (Carbyn 1982).

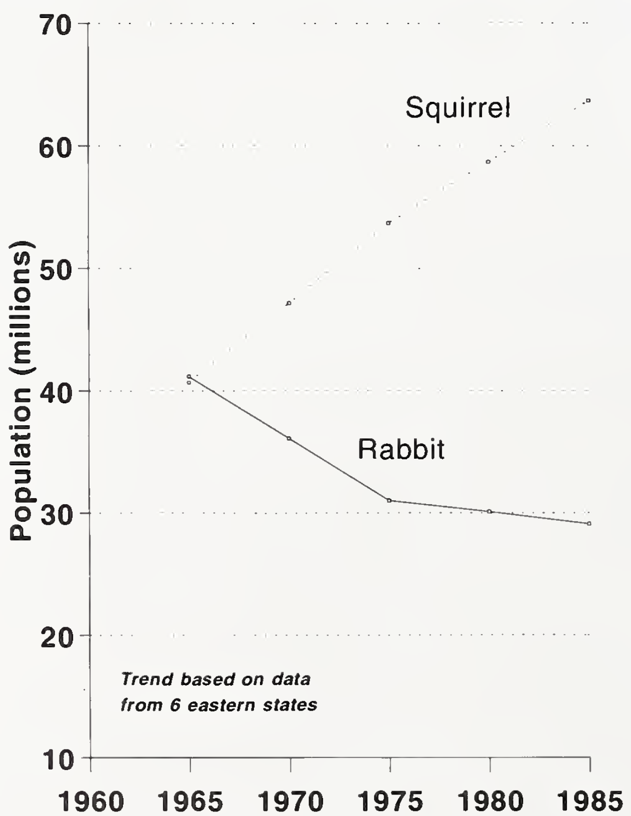

Figure 2a.—Trends in squirrel and rabbit populations, 1960-1985.

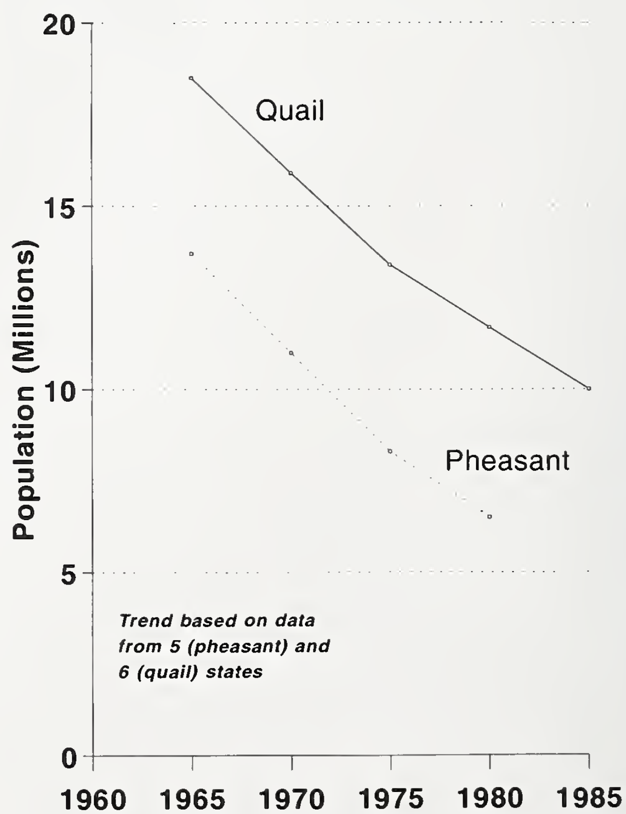

Figure 2b.—Trends in quail and pheasant populations, 1960-1985.

Birds.—Population trends for bird species are more abundant than for any other taxon. Game bird population trends, including upland game birds and waterfowl, vary widely. Quail and sharp-tailed grouse *(Tympanuchus phasianellus)*, both associated with agricultural lands, both showed downward population trends between the 1960s and 1980s (fig. 2)(Flather and Hoekstra 1989). The northern bobwhite *(Colinus virginianus)* showed the greatest and most widespread decline. Over the past decade, bobwhite abundance declined an average of 3.5% per year, with more pronounced declines in the eastern and central U.S.[3] These trends are associated with the removal of habitat in agricultural areas (such as windbreaks and hedgerows) that once provided habitat for abundant populations, and with increases in populations of ground predators (National Research Council 1982). Forest-associated game birds showed no significant trends, except the ruffed grouse *(Bonasa umbellus)*, which increased 3% per year nationally between 1982 and 1991.[3]

Duck populations declined notably since the early 1970s. Breeding populations for 10 species, that collectively comprise 97% or more of the breeding population in the surveyed areas (USDI Fish and Wildlife Service 1974), declined more than 30% from the early 1970s to mid-1980s (USDI Fish and Wildlife Service and Canadian Wildlife Service 1986b). After peaking at about 44 million birds in 1972, populations dropped to a record low of approximately 28 million in 1985. The two most abundant species of duck, the mallard *(Anas platyrhynchos)* and northern pintail *(Anas acuta)*, showed significant historical declines. Recent estimates indicate that breeding duck populations increased to their highest levels since 1986. Despite recent increases, duck abundance remained about 8% below the long-term (1955-1991) mean. Wintering duck populations have shown similar trends (fig. 3). Although mallard populations increased significantly from 1991-1992, they remained 17% below the long-term mean. Other species that remained well below the long-term mean include the northern pintail (-54%), scaup (*Aythya* spp.) (-17%), American wigeon *(Anas americana)* (-16%), and canvasback *(Aythya valisineria)* (-10%). The northern pintail population estimate fell to a record low in 1991. Of the 10 most prominent duck species, only the gadwall *(Anas strepera)* has substantially increased over time (+57%) (USDI Fish and Wildlife Service and Canadian Wildlife Service 1992).

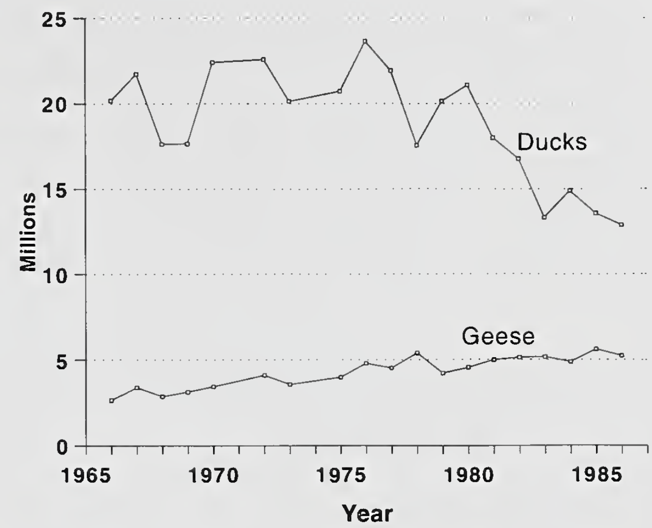

Figure 3.—Trends in wintering duck and geese populations, 1965-1985.

Trends in wintering continental Canada goose *(Branta canadensis)* populations generally have been more favorable than for ducks (fig. 3). Goose populations have gone from an average of 3 million during 1966-1969 to an average of 5.2 million during 1982-1985. Exceptions to this include the Aleutian, *(Branta canadensis leucapareia)*, cackling *(Branta canadensis minima)*, and dusky *(Branta canadensis occidentalis)* subspecies of Canada goose, which declined because of reduced habitat, hunting, and natural disturbance (Amaral 1985, Butler 1985, Cline and Lenhart 1985). Atlantic Canada geese were marked by a record low population index in the 1990-1991 winter survey (USDI Fish and Wildlife Service 1992). Swan populations substantially recovered in the past several decades. Tundra swans *(Cygnus columbianus)* and trumpeter swans *(Cygnus buccinator)* both have slow, but consistently upward trends. None of the three subpopulations of trumpeter swans now are considered to be in danger of extinction (USDI Fish and Wildlife Service and Canadian Wildlife Service 1986a).

Woodcock *(Scolopax minor)* breeding population indices were relatively stable from the mid-1960s to mid-1980s. The breeding index reached a record low during the 1982-1984 period, but the indices have since recovered. The mourning dove *(Zenaida macroura)* is one of the most abundant birds in North America. Nationally, breeding populations gradually declined between 1966 and 1987, reaching a low

[3] USDI Fish and Wildlife Service, Office of Migratory Bird Management, unpublished data.

of approximately 75% of the 1966 breeding population in 1984 (Dolton 1987). Both woodcock and mourning dove populations have stable trends for the past decade (Dolton 1992, Straw 1992).

Native, endemic grassland birds declined in the past 25 years more consistently, and across a broader geographic range than any other group of birds. Only 12 bird species are considered endemic to grasslands; 25 other species are considered highly associated with grasslands. Across all grassland species, only two have increased significantly since 1966, while four endemics and six associated species had statistically significant population declines. Thirteen additional grassland species have downward population trends. The overall trend has been the decline of native, endemic species, while alien and exotic species have colonized newly created forest patches created by fragmentation, fire control, and woody plant invasions (Knopf, In press).

The Breeding Bird Survey (BBS), conducted by the Fish and Wildlife Service, provides information on populations trends of breeding birds in the United States and southern Canada. An analysis of the data from 1966-1991 for the species with adequate sample size indicated that 19% of bird species sampled had increasing populations, 20% had decreasing populations, and 61% were stable[4] (fig. 4). Although most breeding bird populations appear to be stable, a significant number of species have declining populations. Neotropical migratory birds are of special concern. Long-term survey information for the eastern U.S. indicated that populations of 71% of neotropical migrant species declined between 1978 and 1987. However, of the 44 species showing negative trends, only 20 were statistically significant (Robbins et al. 1989).

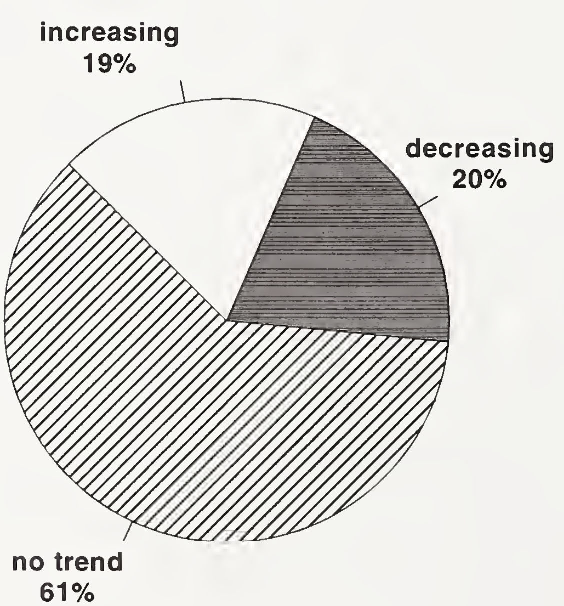

Figure 4.—Trends in breeding bird species, 1966-1991, based on 348 species with adequate sample.

More descriptive data on trends was available for a few other groups of avian species: colonial wading birds, shorebirds, and raptors. Colonial wading birds and shorebirds were seriously depleted in the early 1900s, by commerical exploitation. Most species of colonial wading birds have substantially recovered and have remained stable or even expanded their ranges. The major exception is the wood stork *(Mycteria americana)*, whose population declined so low, it received federal protection as an endangered species. Most species of shorebirds also have made substantial recoveries (with the exception of a few federally listed species). However, their vast ranges make populations difficult to track. They are considered highly vulnerable, because of their reliance on coastal marshes and wetlands for most of the year, which are habitats that have been prone to conversion and degradation (Council on Environmental Quality 1986).

Evans (1982) evaluated the status of 12 raptor species characterized by either recent population declines or inconclusive evidence of population change. The bald eagle *(Haliaeetus leucocephalus)*, osprey *(Pandion haliaetus)*, peregrine falcon *(Falco peregrinus)*, Cooper's hawk *(Accipter cooperii)*, merlin *(Falco columbarius)*, and sharp-shinned hawk *(Accipter striatus)* responded favorably to restrictions on the use of organochlorine pesticides, although the first three species remain on the federal threatened and endangered species list. The crested caracara *(Caracara plancus)*, burrowing owl *(Athene cunicularia)*, and northern aplomado falcon *(Falco femoralis septentrionalis)* were continuing to decline over their ranges, primarily because of lost critical habitat elements. The status of the ferruginous hawk *(Buteo regalis)*, northern harrier *(Circus cyaneus)*, and prairie falcon *(Falco mexicanus)* was unclear. A notable change in raptor trends is the decline in sharp-shinned hawk

[4] *Data provided by USDI Fish and Wildlife Service, Office of Migratory Bird Management.*

populations in the past decade. Population declines ranging from 25% to 75% have been observed in hawk migration counts, along the eastern coastal U.S. The cause of the decline is unknown (Paul Kerlinger, Cape May Bird Observatory, personal communication).

Fish.—Few population estimates exist for fish species. Trend information on distribution and abundance are available from specific regional studies. Anadromous salmonid stocks have been studied in several regions. Beland (1984) estimated that, in precolonial times, the Atlantic salmon *(Salmo salar)* population included up to 500,000 returning adults migrating up 34 New England river systems. The Fish and Wildlife Service (1984) estimated that 7,000 adult salmon now enter only 16 New England river systems. Furthermore, only 1,000 of the returning adult spawners were from natural reproduction; the remainder were from hatchery stock. Salmonids in the Columbia River basin decreased from an estimated 12-16 million individuals in the 1880s to 2.5 million in the 1980s (Meffe 1992). Trends since 1965 for chinook salmon *(Oncorhynchus tschawytscha)* indicate that lower-river (i.e., below Bonneville Dam) chinook runs significantly improved because of increased hatchery production, while upper-river runs declined sharply. Again, the combination of hydroelectric development, commerical harvest, and habitat degradation appear to be major causes of the decline (Phinney 1986). However, hatchery-based management also has been implicated in the decline and extinction of Pacific salmon stocks (Wright 1993).

Some resident salmonids also have experienced range restrictions and population declines. In the Appalachian region of Tennessee, brook trout *(Salvelinus fontinalis)* occupied only 20% to 30% of their estimated range at the turn of the century (Bivens et al. 1985). Severe range restrictions and population declines also have been noted in many native western trout species (Behnke and Zarn 1976). Hybridization and competition with nonnative salmonids have contributed to the decline in both eastern and western trout populations, as has habitat degradation from human developments. A more recent study documented a 45% increase in the number of rare North American freshwater fish during the past decade (Williams and Rinne 1992).

Amphibians.—Data on trends in amphibian populations are largely anecdotal. However, documented declines in local amphibian population have been so widespread in the past decade, both in the U.S. and worldwide, that scientists believe that some amphibian species are declining at rapid rates, some to the point of apparent extinction (Barinaga 1990, Phillips 1990). Available data indicate regional variations in the rate of decline; the most extensive declines appear to be in the northwestern U.S., while the southeastern U.S. appears to have limited declines (Wyman 1990).

Status and Trends in Community and Ecosystem Diversity

Community and ecosystem diversity is concerned with both the composition (areal extent) and configuration (spatial arrangement) of ecosystem types over the landscape.

Ecosystem Composition

One of the few comprehensive and spatially extensive descriptions of terrestrial and wetland ecosystems across the United States is Küchler's Potential Natural Vegetation (PNV) types (Küchler 1964). Klopatek et al. (1979) used the Küchler PNV types to compare existing vegetation to vegetation that would exist if human intervention had not occurred, for data reflective of 1967. On a state-by-state comparison, the loss of natural vegetation ranged from 4% in Nevada to 92% in Iowa. Tule marshes in California showed the greatest proportionate loss (89% reduction), followed by elm-ash forests (88%), bluestem prairie (85%) and sand pine scrub (85%). Once, 23 PNV types covered more than half of the lower 48 states; now, only 13% is covered by those types. A reanalysis of natural vegetation in the conterminous U.S., using the 1982 National Resources Inventory (U.S. Department of Agriculture, Soil Conservation Service 1987) indicated that Florida lost an additional 12% of natural vegetation since 1967; Arkansas, Louisiana, and Missouri each lost an additional 7% (Flather, unpubl. data).

Although Küchler's classification includes wetland systems, the classification has been criticized for underrepresenting wetland types (Burgess 1965, Klopatek et al. 1979). However, a more detailed analysis of the status and trends in wetlands is possible, because this ecologically important system has been the focus of several extensive inventories.

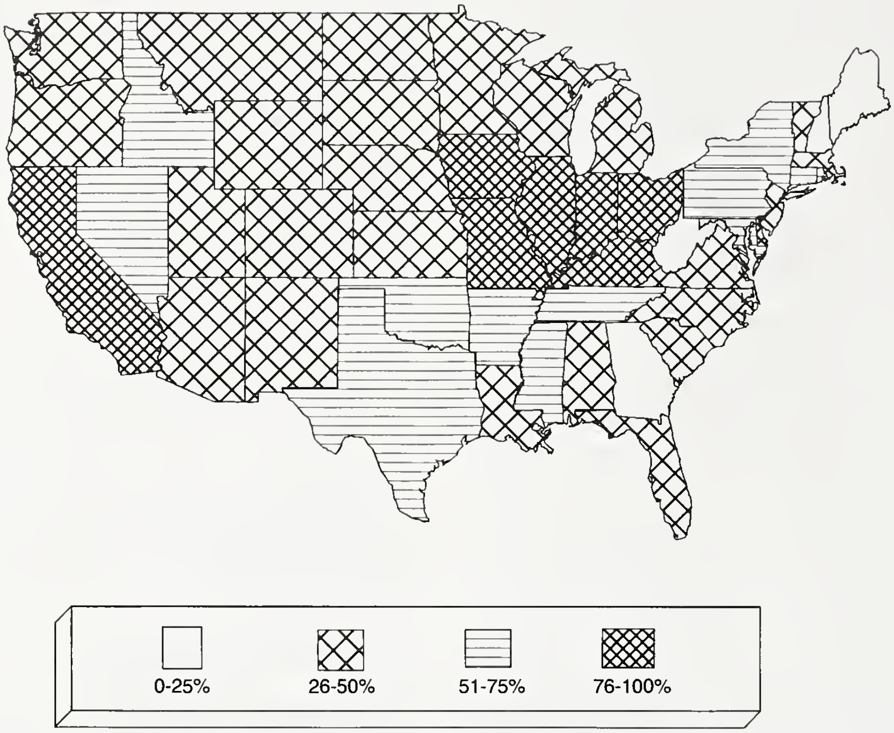

Figure 5.—Percent wetland losses in the conterminous U.S., 1780s-1980s.

Wetlands in the conterminous U.S. have declined from 221 million acres (circa 1780s) to 104 million acres, representing a 53% reduction as the surface area of the lower 48 has gone from 11% to 5% wetland (Dahl 1990). A total of 22 states have lost more than 50% of their wetlands, and 10 states (Arkansas, California, Connecticut, Illinois, Indiana, Iowa, Kentucky, Maryland, Missouri, and Ohio) have lost more than 70% of their original wetland area (fig. 5).

The rate of wetland loss has declined over time, as the wetland area base that could be feasibly drained has declined. From the mid-1950s to the mid-1970s, approximately 460,000 acres were lost annually (Frayer et al. 1983). From the mid-1970s to the mid-1980s, that rate had declined to 290,000 acres per year (Dahl and Johnson 1991). A recent examination of wetland trends from 1982-1987 indicates a continuation of this trend, with 180,000 acres per year being converted to nonwetland uses over the 5-year period (Brady and Flather 1994).

Although the proportionate loss of wetland systems has been considerable, the most extensive land use change (in absolute area) was the conversion of native forest and grassland communities to agriculture. Cropland accounts for 421 million acres, and pastureland covers 132 million acres of the Nation's land base (USDA Soil Conservation Service 1989), totaling 553 million acres that once were in native forest, grassland, and wetlands. These conversions occurred primarily in the eastern and midwestern U.S. More than 70% of the maple-basswood and beech-maple forests of the northern U.S. were converted to cropland, pasture, or urban use by 1972; more than 85% of the prairies dominated by bluestem grass were converted to pasture and cropland (Klopatek et al. 1979). Large areas of oak savanna and prairie ecosystems were converted in Indiana, Illinois, and Iowa. The scale of these changes can be demonstrated by statewide changes in Illinois. Illinois had approximately 38 million acres in forest and prairie in 1800. By 1900, only 8 million acres remained in native ecosystems; only 28% of the forestland and 17% of the native prairie remained (Burger 1978).

Grassland ecosystems have been diminished to a greater extent than forest ecosystems, particularly the prairies of the Great Plains. The tallgrass prairie of the eastern Great Plain is virtually gone. Tall grass prairie once covered more than 1 million km^2 in the Midwest; today less than 1% remains in natural vegetation (Keystone Report 1991). The shortgrass prairie of the western Great Plains (Colorado, Mon-

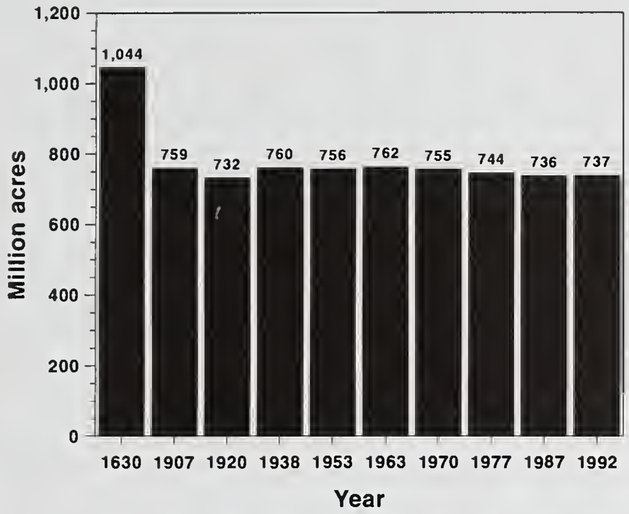

Figure 6.—Trends in forest area in the conterminous U.S., 1630-1992.

tana, and Wyoming) has been fragmented, but prairie can be found on the 15,436 km^2 in National Grasslands, which are primarily shortgrass prairie (Knopf 1993). Other grassland ecosystem losses in the Great Plains have been estimated based on Küchler PNV types, including a loss of 65% of the bluestem-grama ecosystem, 45% of the grama-buffalograss, and 6% of Nebraska sandhills prairie (Klopatek et al. 1979).

Total forest area has declined by about 308 million acres since 1600 (fig. 6). Forests now cover about 737 million acres, two-thirds of the original extent of forests. The existing eastern forest is largely a result of the reversion of agricultural land back to forestland (Powell, et al. 1993). Although forest area has remained relatively stable since the 1920s, specific forest communities have been reduced significantly. Eighty percent of the bottomland hardwoods of the lower Mississippi River delta have been converted to other land uses. Riparian forest in the arid and semiarid West are considered the most modified land type in the West (Tiner 1984). Notable area declines (i.e., greater than 10% reductions from 1963-1987) in southern pines, aspen-birch, and elm-ash-cottonwood have been observed since the early 1960s (Flather and Hoekstra 1989). Mature and old-growth softwood stands are becoming increasingly rare in the major timber-producing regions of the Pacific Northwest and South.

The successional stage and the composition of both plant and animal species on the remaining forest and grasslands has been affected by historical and current management regimes. Fire suppression in the eastern parts of the prairie ecosystem has allowed encroachment of trees and shrubs from the eastern deciduous forest. Supression also has allowed invading species, such as Kentucky bluegrass *(Poa pratensis)* and smooth brome *(Bromus inermis)*, to replace native grasses. The introduction of exotic plant species has significantly impacted grassland communities, displacing native plant species and often reducing the ability of the grasslands to support domestic or wild herbivores. For example, diffuse and spotted knapweed (*Centaurea diffusa* and *C. maculosa*) have infested more than 3.5 million acres in Oregon, Washington, Idaho, and Montana. Leafy spurge *(Euphorbia esula)* covers 2.5 million acres in North America. Yellow starthistle *(Centaurea solstitialis)* has been reported in 23 of the lower 48 states; more than 7.4 million acres have been infested in California alone (Joyce 1989).

The distribution of forest lands by successional stage also has been affected by management. For example, an assessment of the east-side forests of Washington and Oregon indicated that, over the past 40-55 years, the number of acres in early-seral, late-seral, and climax stages decreased, while the acreage in mid-seral stages increased (USDA Forest Service 1993). In the eastern hardwood forests, disturbance (human or other) has not kept pace with forest growth, resulting in greater area in older hardwood stands, in the North.

Aquatic ecosystems also have been impacted by human development. The concentration of human populations around rivers, lakes, and oceans increases the proportionate impact on aquatic areas. Physical alteration of aquatic ecosystems by channelization, construction of dams and reservoirs, and conversion of wetlands, has reduced the total area in aquatic habitat, as well as changed the nature of remaining aquatic habitats. At least 20% of the Nation's 1 million stream miles are estimated to have been modified, including an estimated 5% net loss of length in major streams nationwide. Large reservoirs (> 500 surface acres) have inundated approximately 15,000 miles of streams (Brinson, et al. 1981) Reservoirs provide habitat for a totally different group of species than do free-flowing streams. Exotic species often are introduced to replace the native fauna. In addition, dams block access of migratory fish to upstream areas, reduce downstream flows, and reduce nutrient flows (Moyle and Leidy 1992).

Land use intensification also indirectly impacts aquatic ecosystems. Land use alterations adjacent to waterways, and runoff from agricultural and urban land also have affected extensive areas of aquatic environments (Allan and Flecker 1993). Perhaps the most telling assessment of biological diversity in aquatic systems is Benke's (1990) finding that only 2% of the streams in the contiguous U.S. retained sufficient natural high qualities to warrant federal protection as wild, scenic, and recreational rivers.

Ecosystem Configuration

Information from remote sensing provides data on trends in the configuration of ecosystem types. Much of what is generalized from the literature is based on an aggregation of local studies. Patches of natural habitats in the eastern United States have become smaller and increasingly isolated. Historical accounts, such as those reviewed by Burgess and Sharpe (1981), illustrate the extent to which forest ecosystems have been diminished and isolated over time. Similar patterns have been described for the timber-producing regions of the western U.S. (Hanson et al. 1991, Ripple et al. 1991). A spatially extensive analysis of landscape pattern indicates that highly dissected landscapes are found throughout the eastern U.S. (O'Neill et al. 1988). However, depending on the time period examined and the geographic location, other patterns can emerge. Turner (1990a) noted that land-use change in rural Georgia allowed forest ecosystems to become less fragmented during the period 1930-1980.

One of the issues that brought attention to the importance of landscape configuration was mounting evidence that the fragmentation of natural habitats poses serious survival problems for certain types of species. Particularly vulnerable to fragmentation are species that have large home range requirements, such as large carnivores; species that do poorly near ecological edges, including many forest interior bird species; and species that have poor dispersal abilities and become marooned on isolated fragments. Fragmentation is seen as the principal threat to most species in the temperate zone (Wilcove et al. 1986).

Studies of birds provide evidence linking fragmentation to species composition. For example, in a detailed study of breeding birds of central Maryland, Whitcomb et al. (1981) observed that forest interior species were rare on small (1-5 ha) forest patches, and abundant on large (>70 ha) forest tracts, accounting for 80-90% of the breeding individuals. If larger forest tracts are lost, these forest interior birds will fare poorly in smaller forests. Additional studies are summarized in Wilcove and Robinson (1990) and Samson (1981).

In an attempt to look at avian diversity patterns across eastern forested ecosystems, Flather et al. (1992) found support for wide-ranging impacts from land-use intensification. They analyzed breeding bird survey data (Droege 1990), in conjunction with land use and land cover data from the National Resource Inventory (USDA Soil Conservation Service 1987) and digitized land cover data from the U.S. Geological Survey (USDI Geological Survey 1987). Avian diversity was reduced (i.e., supported a smaller proportion of the expected species community) as both vertical habitat structure and land type diversity were reduced.

Assessment of the potential influence of landscape changes on diversity in the western U.S. also is alarming. Picton (1979) found that large mammals in the mountain ranges of central Montana had been isolated by human development. A comparison between the historic number of species present and the number of species remaining after development (and before reintroduction of species began), showed that the percent of species remaining was correlated with habitat area. In a direct study of the influence of landscape structure on vertebrate diversity in the Cascade Range of Washington, Lehmkuhl et al. (1991) found that avian richness increased in response to edge effects associated with clearcuts; amphibian richness was strongly associated with land type dominance; whereas mammal richness showed no strong relation with landscape structure. These results suggest that forest fragmentation in this geographic region may not have reached a critical threshold beyond which a decline in biotic diversity would be observed (Lehmkuhl et al. 1991:441).

FEDERAL ROLE IN BIOLOGICAL DIVERSITY

The public lands in federal ownership encompass a large part of natural variety in the United States. Federally owned lands account for 37% of the Nation's land base (Crumpacker et al. 1988). Federal lands provide habitat for a large proportion of the Nation's wild species. The National Forest System (26% of

federal lands) provides aquatic and terrestrial habitat to support more than 3,000 species of wildlife and fish (Flather and Hoekstra 1989). Federal and state lands provide habitat for a large share of threatened and endangered species. Of the 660 listed species for which data were available, the National Forest System lands contained 24% of those species; the Bureau of Land Management lands contained 17%, and Department of Defense lands contained 26% of the species (Flather et al. 1994).

Several studies assessed the representation of ecosystem types on federal lands. The broadest assessment was by Crumpacker et al. (1988), in a study of whether Küchler's Potential Natural Vegetation (PNV) types were adequately represented on federal and Indian lands. If a PNV type did not occur within the boundaries of a protected area, then actual biological samples were considered inadequately protected. Of the 135 PNV types, 9 types were not represented at all. Twenty-four types had limited representation (ranging from 4,003 to 52,539 total hectares). Of those 24, 11 always have been rare or largely converted to nonnatural uses, while 13 generally are common elsewhere in the United States, or are less affected by conversion. Special attention was paid to PNV types that are spatially extensive, but are facing threats of conversion off federal lands. National Forest System lands included 73% of those types, the National Park Service included 67%, and Fish and Wildlife Service lands included 47%.

Kirby (1984) and Davis (1984) used a system of 242 ecosystems that were based on a combination of Bailey's ecoregions and Küchler types to estimate ecosystem coverage on National Forest lands. Kirby (1984) estimated that the National Forests contain more than 105 of 242 ecosystem types in the United States. Davis (1984) assessed the ecosystem coverage within the National Wilderness Preservation System (NWPS). Representation within the NWPS was considered adequate for an ecosystem if two or more distinct examples of at least 1,000 acres each existed. Using this system, 81 of 233 basic ecosystems were adequately represented in designated wilderness. At the time, the NWPS had 80 million acres.

Federal lands contain much of the diversity in the U.S.; and they are uniquely designed to play an important role in conserving biological diversity. Federal lands cover about one-third of the total land base. However, these lands are concentrated in the western U.S.; and many elements of diversity are not represented on federal lands. Therefore, private lands are also critical to conserving diversity. Furthermore, although federal lands often are the sole refuge for species requiring large isolated tracts of land, these lands are not sufficient to protect the long-term viability of some species. Grumbine (1990) concluded that the current configuration of reserved lands (primarily National Parks and National Forests) was not sufficient to ensure the long-term viability of large vertebrates, including the grizzly bear, cougar, wolverine *(Gulo gulo)*, black bear, and gray wolf. The lack of federal lands in the eastern U.S. limits the Federal role in conserving diversity in the eastern U.S. A conservation strategy needs to include land in all ownerships to adequately address the full range of diversity.

SUMMARY AND CONCLUSIONS

Information on status and trends of biological diversity in the United States is limited in comparison to the full array of biological diversity. Data on genetic diversity is limited primarily either to species of commercial value or species considered vulnerable to extinction. Data on trends in species diversity are the most extensive. Yet, even these sources of information are restricted mostly to threatened and endangered species, game species, and avian species. Aside from trends in threatened and endangered species, no comprehensive data were found on the status and trends of reptiles, amphibians, invertebrates, plants, and micro-organisms.

The future of biological diversity also is likely to be confounded by the impacts of climate change. Although there is great uncertainty, individual species are likely to react differently, resulting in different migration rates and new animal and plant communities (Joyce et al. 1990). Both plants and animals are sensitive to climate. Potential changes could include changes in the range of many species, disruptions of natural communities, and extinctions of species incapable of adaptation. Evidence from the fossil record indicates climate change has been an important cause of extinctions in geologic time (Peters 1988).

The lack of consistent, comprehensive data on the range of diversity components shows that better information is needed to both understand and manage biological diversity. There is a substantial literature on approaches to monitoring biological diver-

sity, and the types of indicators that are needed to provide sufficient information (Noss 1990, Salwasser 1990). The major challenge is devising a system that is sufficently complex to capture the range of diversity without being too unwieldy to provide useful management information.

Regardless of the limited data, the evidence is sufficient to conclude that major changes have occurred in the distribution and abundance of biological diversity, in the U.S. Comparison of data from the 1960s indicates that native diversity has declined in the U.S., over the past three decades, and is likely to continue to decline. Although the long-term viability of most of the Nation's plants and animals appears to be secure, almost 800 species and subspecies are already federally protected, and another 3,500 species and subspecies are likely to be listed. Invertebrates and plants, which will dominate the list of species at risk in the future, are critical species in the functioning of ecosystems. Yet relatively little is known about the status or life histories of most invertebrates. Other species, although not in danger of extinction, have been exhibiting declining populations for several decades.

Development of a common set of definitions and indicators for monitoring biological diversity would improve the ability to assess trends over time. However, those trends will reflect land management choices. Conservation of biological diversity must be addressed within managed landscapes, on all ownerships (Hanson et al. 1991, Pimentel et al. 1992). Although no major land use changes are expected that rival the massive shifts from forest and prairie to agricultural land during the 19th and early 20th century, natural habitats will continue to decline and fragment as a result of growing human populations and associated development. The net result will be increasing pressure on a shrinking base of natural habitats.

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Abstract

Langner, Linda L.; Flather, Curtis H. 1994. Biological diversity: status and trends in the United States. Gen. Tech. Rep. RM-244. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 24 p.

Biological diversity in the United States is summarized in three categories: genetic diversity, species diversity, and community/ ecosystem diversity. Major changes have occurred in the distribution and abundance of native diversity, in the United States, in the past 300 years. Native plant and animal diversity has declined over the past three decades, and is likely to continue to decline because of continuing human population growth and associated development.

Keywords: biological diversity, genetic diversity, species diversity, ecosystem and community diversity, threatened and endangered species